ONE HUNDRED SIXTH CONGRESS

Congress of the United States

House of Representatives

COMMITTEE ON GOVERNMENT REFORM
2157 RAYBURN HOUSE OFFICE BUILDING
WASHINGTON, DC 20515–6143

MAJORITY (202) 225–5074
MINORITY (202) 225–5051
TTY (202) 225–6852

August 3, 2000

BY FACSIMILE

The Honorable Carol M. Browner
Administrator
Environmental Protection Agency
401 M Street, S.W.
Washington, DC 20460

Dear Administrator Browner:

I am writing to request that the Environmental Protection Agency (EPA) evaluate recent research on the feasibility of EPA's 1997 National Ambient Air Quality Standard (NAAQS) for ozone, commonly referred to as the "eight-hour" standard.

The Supreme Court recently accepted EPA's petition to review EPA's 1997 NAAQS for ozone and particulate matter. It also accepted a cross-petition from industry lawyers to examine whether the Clean Air Act (CAA) requires EPA to ignore all factors other than the health effects of air pollutants when setting NAAQS. Most prominent among such factors are the cost and feasibility of controlling emissions. EPA estimated that the cost of meeting the NAAQS it issued in 1997 would amount to nearly $50 billion in 2010. However, new research suggests that this estimate is much too low and that the NAAQS as promulgated may simply not be feasible.

Infeasible mandatory standards seem to me contrary to principles of good government. Congress intended the CAA to be "technology forcing," i.e., to pressure regulated sources to develop pollution control devices that might not at the time appear to be economically or technologically feasible. However, there are limits to what even the best new technology can accomplish. Overly-ambitious mandates that require States, communities, and businesses to spend billions of dollars in pursuit of the unattainable run the risk of undermining respect for law and fostering mistrust of the Federal Government. Moreover, standards that deliver little or no air quality improvement at great cost may actually harm public health by suppressing economic growth and by diverting attention, effort, and money from more serious health, safety, or environmental threats.

Therefore, pursuant to the Constitution and Rules X and XI of the United States House of Representatives, I request that EPA evaluate new research on the cost and feasibility of the eight-

hour standard by addressing the questions in the enclosure. I request that EPA deliver its response to the Subcommittee majority staff in B-377 Rayburn House Office Building and the Subcommittee minority staff in B-350 Rayburn House Office Building by Friday, August 25, 2000. If you have any questions about this letter, please contact Subcommittee Staff Director Marlo Lewis at 225-1962. Thank you for your attention to this request.

Sincerely,

David McIntosh

David M. McIntosh
Chairman
Subcommittee on National Economic Growth,
Natural Resources, and Regulatory Affairs

Enclosures

cc: The Honorable Dan Burton
The Honorable Dennis Kucinich

Q1. Does the Environmental Protection Agency (EPA) believe that National Ambient Air Quality Standards (NAAQS), if shown to be infeasible using emissions control technologies that are currently available or expected to be available in the foreseeable future, should nonetheless be promulgated and enforced?

Q2. If EPA currently sets air standards without regard to cost or feasibility, is there anything in the standard-setting process that would prevent EPA from promulgating standards that cannot be attained using emissions control technologies that are currently available or expected to be available in the foreseeable future?

Q3. In an article recently published in *Environmental Science & Technology*, Darrell Winner and Glen Cass develop a new air quality model for the "Los Angeles ozone problem," which they view as a "prototype" for severe photochemical smog problems elsewhere.[1] Their executive summary states that "control of organic vapor and oxides of nitrogen emissions" can reduce the occurrence of peak one-hour ozone concentrations above 0.12 parts per million (PPM) to approximately "20 days per year." (Note: such frequent occurrence of one-hour concentrations above 0.12 PPM would violate the existing ozone standard.) The researchers also conclude that, "even at very stringent levels of emission controls," southern California would exceed the new 1997 ozone standard (0.08 parts PPM over an eight-hour averaging time) "more than 60 days per year." An implication of the Cass-Winner research is that "it may be physically impossible for an airshed to attain compliance with the standards that have been set based on public health considerations" (p. 2617).

a. Does EPA know of technical shortcomings in the Winner-Cass research to suggest that it greatly overestimates the emissions reductions necessary to achieve the 1997 standards?

b. The Winner-Cass study appears to imply that, in southern California, the 1997 ozone standard is infeasible using emissions control technologies that are currently available or expected to be available in the foreseeable future. Does EPA agree?

c. If not, please outline a plan to implement the 1997 ozone NAAQS in southern California that would result in attainment in the foreseeable future.

Q4. Standard procedure in regulatory analysis is to estimate the costs and benefits of full compliance with a new regulation. Full compliance with the 1997 ozone and

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[1]Darrell A. Winner and Glen R. Cass, "Effect of Emissions Control on the Long-Term Frequency Distribution of Regional Ozone Concentrations," *Environmental Science & Technology*/ Vol. 34, No. 12, 2000, pp. 2612-1617.

particulate matter (PM) standards would require that all parts of the country meet the standard regardless of weather conditions.

a. Despite dramatic improvements in air quality over the last three decades, about 92 million Americans still reside in 32 metropolitan areas where pollution concentrations exceed EPA's 1979 one-hour ozone standard. When EPA issued the 1997 NAAQS for PM and ozone, did EPA perform an analysis of the likelihood that the emissions reductions it modeled would still leave some parts of the country out of compliance with those standards?

b. Do the cost estimates that EPA developed in 1997 reflect a set of emissions controls sufficient to assure that all areas of the country will be in full compliance with the 1997 NAAQS? If not, what expectations regarding levels of non-compliance do those cost estimates reflect?

c. Would the costs of emission controls sufficient to bring all areas of the country into attainment with the eight-hour ozone standard be greater than EPA's 1997 cost estimates? If so, how much greater?

5. In its 1997 analysis of the costs of meeting the PM and ozone standards, EPA avoided the conclusion that the ozone standard was infeasible by assuming that any additional emissions reductions needed to meet the standard would be available at a constant cost of $10,000 per ton. However, according to Randall Lutter of the AEI-Brookings Joint Center for Regulatory Studies program, EPA's data suggest that the costs of reducing emissions by one ton rise rapidly with greater emissions reductions.[2] By extrapolating the trends implicit in EPA's data, Lutter concluded that the costs are so high as to suggest the standard will never be met. By relaxing EPA's assumption that control costs are capped at $10,000 per ton, he estimated that the total costs of meeting the standard in just seven cities would reach about $70 billion in 2010. In Fresno, California, where EPA estimated emissions must be cut by more than 60 percent from projected 2010 levels, Lutter calculated that the costs of meeting the standard could reach more than $4.6 trillion dollars.

a. Has EPA assessed how its estimate of the costs of the 1997 ozone standard would vary with alternative assumptions about the incremental cost of emissions reductions beyond those that can be achieved by identifiable control technologies? If so, please provide a copy of this analysis.

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[2] Randall Lutter, "Is EPA's Ozone Standard Feasible?" AEI-Brookings Joint Center for Regulatory Studies, Regulatory Analysis 99-6, December 1999.

b. Is the incremental cost of reducing emissions generally likely to rise with greater emissions reductions, given a fixed set of emission control technologies?

c. Would EPA need to revise its cost estimates for the 1997 ozone standard if EPA were to use the relationship between cost and emissions reductions from identifiable technologies as a basis for extrapolating the costs of emissions reductions beyond those achievable by identifiable control technologies? If so, please provide EPA's revised estimates of the costs of meeting the 1997 standard

Environ. Sci. Technol. 2000, 34, 2612–2617

Effect of Emissions Control on the Long-Term Frequency Distribution of Regional Ozone Concentrations

DARRELL A. WINNER AND
GLEN R. CASS*

Environmental Engineering Science Department, California Institute of Technology, Pasadena, California 91125

Photochemical airshed models that simulate the pollutant transport and atmospheric chemical reaction processes leading to ozone formation now can be exercised for years at a time, permitting a thorough evaluation of the extent to which urban and regional ozone concentrations can be controlled. The Los Angeles ozone problem serves as a prototype for severe photochemical smog problems elsewhere. In southern California, the occurrence of peak 1-h average ozone concentrations above 0.12 ppm can be reduced to approximately 20 days per year through control of organic vapor and oxides of nitrogen emissions. Calculations show that the number of days per year with 1-h average O_3 concentrations above 0.12 ppm approaches zero more quickly in response to controls than is the case for the number of days with lower but more persistent ozone concentrations; as a result, more than 60 days per year will exceed the new U.S. Federal ozone standard set in 1997 at a level of 0.08 ppm over an 8-h averaging time, even at very stringent levels of emission control. The days with the highest observed ozone concentrations are not necessarily the hardest days to bring below the air quality standards.

Introduction

Increased ozone concentrations in the lower atmosphere can adversely affect human health as well as damage forest systems, reduce agricultural yields, and degrade sensitive materials (*1*, *2*). For these reasons, limits on ozone concentrations have been set by governments around the world. The ozone control program in the United States alone entailed annualized compliance costs of approximately $26.1 billion per year as of 1990 (*3*). Yet in 1999, 92.2 million people in the United States lived in the 32 metropolitan areas where pollutant concentrations still exceeded the Federal ozone standard set at 0.12 ppm O_3 not to be exceeded more than 1 day per year averaged over 3 years (*4*). An even larger number of locations are out of compliance with the new U.S. Federal air quality standard set in 1997 at 0.08 ppm O_3 over an 8-h averaging time calculated as the 3-year average of the annual fourth-highest daily maximum. This new standard cannot be enforced at this time due to a recent court decision in *ATA vs EPA*, No. 97-1441 (D.C. Cir, May 14, 1999). Recalcitrant ozone control problems also exist in parts of western Europe, Greece, Mexico, Brazil, and other developing countries. Clearly, there is still much to be learned about how to control regional ozone concentrations.

* Corresponding author present address: School of Earth and Atmospheric Sciences, Georgia Institute of Technology, Atlanta, GA 30332-0340; e-mail: cass@eas.gatech.edu.

One key prerequisite to finding effective approaches to meeting ozone air quality objectives is understanding the relationship between the reactive organic gases (ROG) and oxides of nitrogen (NO_x) emissions that lead to ozone formation. The effect of emission controls on ozone formation can be represented quantitatively using ozone isopleth (lines of constant concentration) diagrams (*5*–*8*). In a traditional graph of this type, the horizontal axis represents the air basin-wide percentage control of the anthropogenic emissions of reactive organic gases, and the vertical axis represents the air basin-wide percentage control of the emissions of nitrogen oxides. Contour lines connecting the many alternative ROG and NO_x emissions pairs that could be used to reach the same peak 1-h average ozone concentration on the day studied are drawn at several ozone concentrations and over all possible levels of ROG and NO_x emissions. Diagrams of this type that are drawn based on smog chamber experiments, analysis of ambient air quality data, or air quality modeling studies usually show that ozone concentrations can be reduced by starving the atmosphere for either ROG or NO_x, or both. At relatively high ROG/NO_x ratios, ozone formation is limited by the availability of NO_x since there is an ample supply of hydroperoxyl (HO_2) and organic peroxy (RO_2) radicals to convert nitric oxide (NO) to nitrogen dioxide (NO_2); the ratio of NO_2 to NO and solar intensity determine the resulting ozone concentrations. Within this domain, lowering NO_x emissions will result in lower peak ozone concentrations. At relatively low ROG/NO_x ratios, ozone formation is ROG-limited since radicals are scavenged by the high amounts of NO_x present. When NO_x concentrations are lowered, more of the radical pool can react with ROG thereby promoting additional radical formation resulting in higher ozone concentrations. Within this region of an ozone isopleth diagram, lowering ROG emissions alone or possibly even increasing NO_x emissions can result in a decrease of ozone concentrations. Historically, ozone control efforts in southern California have emphasized reducing ROG emissions since the technology was initially less expensive and more readily available, while in rural areas of the eastern United States with its relatively higher biogenic organic vapor emissions from forests NO_x control may be favored (*9*).

Millions of pieces of emissions and meteorological data can be needed to describe the events leading to regional ozone formation for even a single day. Because of this complexity, the almost universal practice has been to model the effect of emission controls on atmospheric ozone concentrations for one or a few historically observed high ozone events when evaluating proposed emission control programs. Yet in the region surrounding Los Angeles in a typical year more than 100 days per year exceed the new Federal 8-h ozone air quality standard; each day must be brought under control. For a secondary pollutant like ozone that is formed in the atmosphere by nonlinear chemical reactions that depend as much on pollutant precursor ratios as on the amount emitted, it is not at all obvious that the days with the highest observed concentrations historically are the hardest ones to bring below the air quality standard. A more thorough investigation of the overall response of the atmosphere to various approaches that might be used to lower regional ozone concentrations is needed.

Recently, methods have been demonstrated for modeling the long-term frequency distribution of daily ozone concentrations using a chemically explicit Eulerian photochemical airshed model (*10*). The procedure involves automated generation of the day-specific emissions inputs and meteorological fields needed to drive the CIT photochemical airshed

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model (*11, 12*) for a period of 1 year or longer. Here we use this modeling approach to examine how the frequency of occurrence of days with O_3 concentrations above specified levels will respond to changes in ROG and NO_x emissions. These findings are cast into a format similar to ozone isopleth diagrams but where contours of constant air quality are labeled according to the number of days per year with ozone above a specified concentration at any selected combination of air basin-wide ROG and NO_x emissions. Methods are illustrated by application to the greater metropolitan Los Angeles area, and scientific findings relevant to ozone control in southern California are discussed. Since ozone air quality control practices in many places around the world are adapted from experience gained in Los Angeles, these findings have importance that extends beyond the local situation in southern California.

Approach

Model Description. In the present work, the model for the frequency distribution of long-term average ozone concentrations developed by Winner and Cass (*10*) is used to determine and display how the number of days per year exceeding specified ozone concentrations responds as a function of the air basin-wide level of ROG and NO_x control. The technical vehicle for performing these calculations is the CIT photochemical airshed model (*11,12*). This Eulerian photochemical model numerically solves the atmospheric diffusion equation:

$$\frac{\partial c_i}{\partial t} + \nabla(\bar{u}c_i) = \nabla(K\nabla c_i) + R_i + S_i \qquad (1)$$

where c_i is the ensemble mean concentration of species i, $\bar{u}$ is the mean wind velocity at location $\bar{x}$ at time t, K is the eddy diffusivity tensor, R_i is the chemical reaction rate of species i that depends on the pollutant species $c_1 \ldots c_n$ and on the temperature T, and S_i is the elevated source emission rate of species i at location $\bar{x}$ at time t.

A no-flux boundary condition is applied at the top of the modeling region. The boundary condition at the earth's surface requires that the upward flux of each chemical species equals the ground-level emissions minus the dry deposition flux:

$$-K_{zz}\frac{\partial c_i}{\partial t} = E_i - v_g^i c_i \qquad (2)$$

where K_{zz} is the vertical eddy diffusivity, E_i is the ground-level emission flux of species i, and v_g^i is the deposition velocity for species i.

The version of the CIT airshed model used here employs a revised dry deposition module based on surface resistance values (*13*) and an extended version of the LCC chemical mechanism (*14*). The LCC chemical mechanism represents organic gas emissions through eight lumped organic species classes: C4+ alkanes, ethene, C3+ alkenes, monoalkyl benzenes, di- and trialkyl benzenes, formaldehyde, C2+ aldehydes, and ketones. The LCC mechanism has been extended by Harley et al. (*11*) to explicitly include the chemistry of methane, methanol, ethanol, methyl *tert*-butyl ether (MTBE), isoprene, hydrogen peroxide, and sulfur dioxide. This extended mechanism contains 35 differential species, 10 steady-state species, and 107 chemical reactions.

CIT Airshed Model Performance Evaluation. A year-long simulation of historical conditions during the calendar year 1987 has demonstrated that it is possible to successfully model ozone concentrations in southern California using routinely measured air quality and meteorological data (*10*). The year 1987 is chosen as an historical baseline both because extensive research has been performed to improve the emissions inventory for that year and because that year contains meteorological conditions that serve as the basis for the proposed 1997 implementation plan for air quality improvement in the Los Angeles area. When supplied with the best estimates of pollutant emissions in the southern California modeling region shown in Figure 1 with motor vehicle emissions scaled upward to match the 1987 Van Nuys Tunnel study (*11, 15, 16*) and calculated according to daily temperature variations, the model predicts 1-h average O_3 concentrations for the entire year 1987 that display +15% normalized bias and an average station peak prediction accuracy of +14%. The region-wide peak O_3 concentration for each day is on average 7% above the observed value over the entire year. For 8-h average O_3 concentrations for the entire year 1987, the model simulations display a +26% normalized bias and an average station peak prediction accuracy of 14%. The predictions for O_3 precursors show a normalized bias of +23% for total NO_2; daily ROG data are unavailable but when such data are available for a few days during the Southern California Air Quality Study (SCAQS) episodes that year, model performance for lumped (and even individual) ROG is favorable (*12, 16*). These quantitative measures of airshed model performance over the course of the year-long simulation of historical conditions are comparable to those normally obtained from simulations of 2 or 3 days using hand-crafted meteorological inputs based on expensive special field measurement programs (*10*).

Emission Inventory. The baseline 1987 southern California emissions inventory for the present study is the same as for the baseline model evaluation study of Winner and Cass (*10*). To minimize the effect of upwind boundary conditions on O_3 concentration predictions in the Los Angeles area, the modeling region used for the present study is greatly extended when compared to previous studies, as shown in Figure 1. San Diego and Santa Barbara counties are now appended to the modeling region along with large areas of the Pacific Ocean. The emissions inventory is based on the 1987 emission inventory developed for the South Coast Air Basin (SoCAB) and adjacent areas (Los Angeles, Orange, Riverside, San Bernardino, and Ventura Counties) by the California Air Resources Board (CARB) plus separate emissions inventories for San Diego and Santa Barbara Counties that have been adjusted to 1987 conditions. In the present study, emission data for motor vehicles are employed that both reflect the effect of hourly changes in ambient temperature on emissions and are scaled upward to match the emission rates measured for on-road operation of the southern California vehicle fleet during the Van Nuys Highway Tunnel study (*11, 15, 16*). Motor vehicle emissions for the SoCAB as a function of time and location throughout 1987 are computed using the Caltrans model DTIM2 (*17*) in conjunction with motor vehicle emission factors from the CARB motor vehicle emission factor program (EMFAC7F) and the hourly, day-specific surface temperature fields. At the conclusion of this process, the emissions over the entire 450 km by 225 km grid of 5 km by 5 km cells covering the area mapped in Figure 1 total 2924 t day^{-1} ROG as compared to 2304 t day^{-1} ROG for the smaller modeling region used previously by Harley et al. (*11*; the above comparison is for the temperatures of August 28, 1987). Speciation of the ROG emissions within the model follows the revised source composition profiles previously compiled by Harley et al. (*18*).

Scale factors adapted from the seasonally resolved biogenic emissions inventory for the South Coast Air Basin reported by Benjamin et al. (*19*) are used to modify the SCAQS August 1987 biogenic hydrocarbon inventory. The scale factors are 0.30, 0.63, 1.00, and 0.40 respectively for winter (January–March), spring (April–June), summer (July–October 10), and fall (October 11–December). Note that

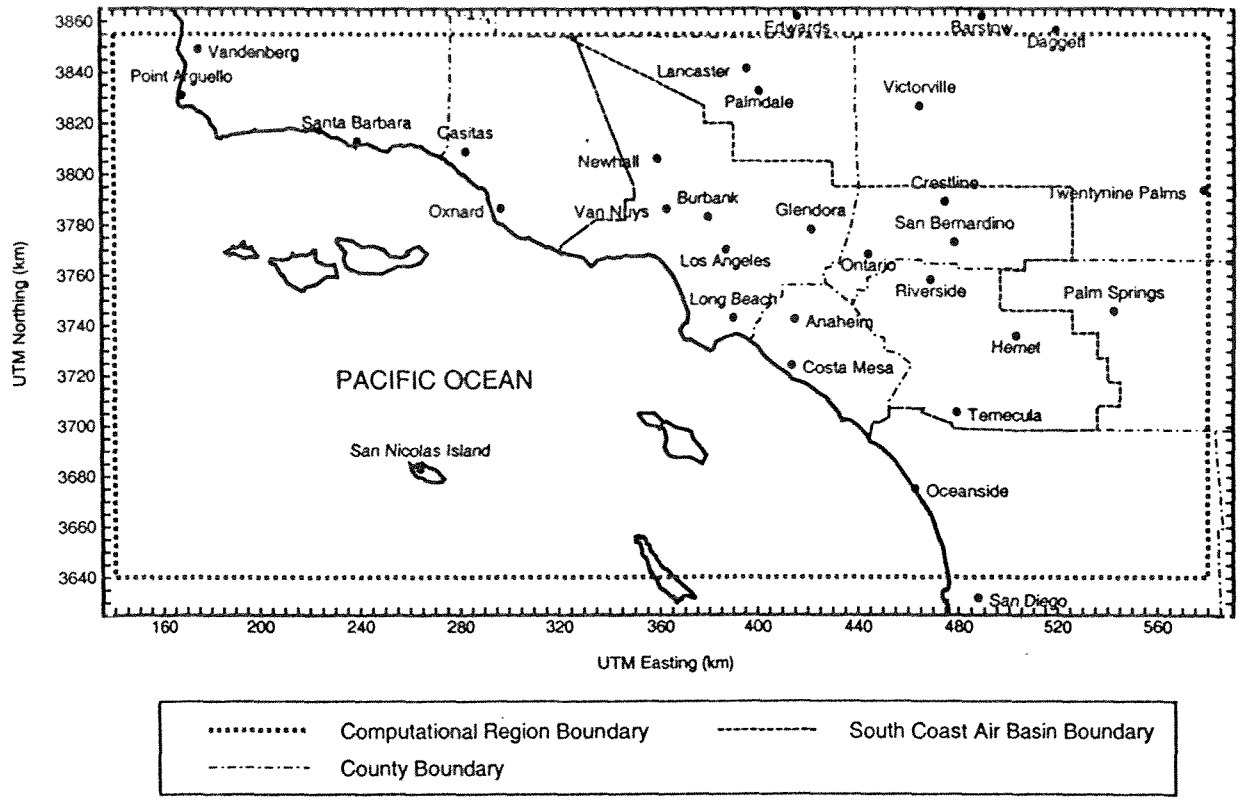

FIGURE 1. Southern California, showing the computational region for the air quality model, the legal boundaries of the South Coast Air Basin that forms the air quality control region for Los Angeles, and the county boundaries.

October 1–10, 1987, was a period of high ozone concentrations and temperatures. The summer scale factor is used for this period.

The total base case ROG emissions within the modeling domain of Figure 1 for the 22 highest emission days of the year 1987 ranged from 3527 to 2986 t day^{-1}. The ROG emissions at a very high level of control with only 15% of base case anthropogenic ROG plus all biogenic emissions remaining, ranged from 635 to 515 t day^{-1} for the same 22 days. These highest emission days occur as early in the year as April 15 and as late as October 5.

Meteorological Input Fields. Input data fields describing temperature, humidity, solar radiation, mixing depth, and wind speed and direction are the same as utilized in the base case model evaluation study (*10*). These fields are generated from historical observations using a second-degree polynomial fitting procedure with an r^{-2} weighting scheme (*20*). The 3-D wind fields are generated using an objective analysis program (*21*) that combines horizontal winds aloft measured daily at six sites up to 1500 m above ground level with surface-level horizontal winds measured hourly at 70 sites. The vertical winds are computed from continuity considerations by the objective analysis program. The mixing depth fields are calculated hourly according to Holzworth's (*22*) method from the interpolated surface-level air temperature field plus the air temperature versus altitude data from the upper air soundings.

Boundary and Initial Conditions. To capture the possible effect of emission controls both inside and outside the modeling region on alternative ROG and NO_x control strategies, an interpolation scheme was developed that sets initial conditions as well as boundary conditions along all edges of the model and at all elevations by interpolation between the historically measured conditions (*10*) and Pacific Ocean clean air conditions (*23*) in proportion to the amount of emission control that is applied within the modeling domain. In effect, this simulation assumes that adjacent jurisdictions reduce their emissions by approximately the same amount as the reductions that occur within the modeling region. NO_x species (NO, NO_2) boundary and initial conditions are interpolated in proportion to the percentage change in NO_x emissions. ROG species boundary and initial conditions are interpolated in proportion to the percentage change in anthropogenic ROG emissions. Ozone boundary and initial conditions are interpolated in proportion to the average of the percentage NO_x and anthropogenic ROG control. Thus, at 100% of 1987 actual emission levels for NO_x and ROG (no control), the boundary and initial conditions are set at the historically observed levels while at 5% of the historical anthropogenic emission levels for NO_x and ROG (maximum control studied), the boundary and initial conditions approach but do not quite reach the Pacific Ocean clean air values.

Isopleth Diagram Generation Technique. A massively parallel supercomputer is used to simulate the effect of 64 different combinations of NO_x and ROG emission levels on hourly ozone concentrations in southern California for the calendar year 1987 meteorological conditions, and the results are contoured to create diagrams that display the number of days above specified ozone concentrations at all levels of emissions control. The emissions levels studied form a matrix of all pairwise combinations of 0, 15, 30, 45, 60, 75, 85, and 95% ROG and NO_x control. Two weeks of CPU time using 64 nodes of a Cray T3E-600 computer are employed to generate these 64 year-long calculations that involve 23 680 days of simulation when the model startup period is considered.

Results

The CIT photochemical airshed model just described is run continuously for meteorological conditions that match every hour of calendar year 1987 over the modeling region of Figure 1. The ability of this simulation to reproduce the time series of 1- and 8-h average O_3 concentrations for every hour of the

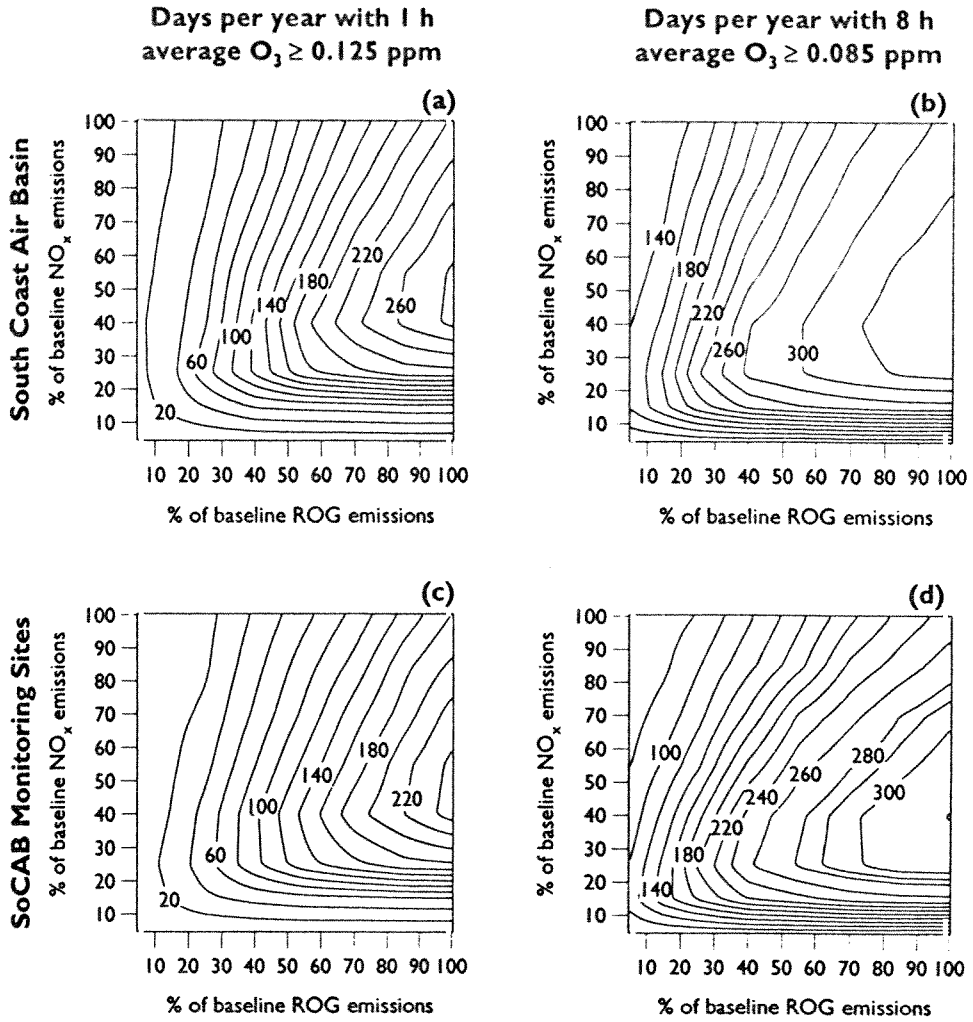

FIGURE 2. Isopleth diagrams showing the predicted number of days that equal or exceed the concentration level of the 1-h (panels a and c) and 8-h (panels b and d) ozone standards as a function of basin-wide anthropogenic ROG and NO_x emission reductions in southern California that are stated as a percentage of 1987 baseline emissions remaining after control. Panels a and b are based on predicted peak concentrations anywhere within the SoCAB; panels c and d are based on values predicted at SoCAB air monitoring sites.

year is discussed briefly in the CIT Airshed Model Performance Evaluation section of this paper and is illustrated in detail in ref *10*. The model is then rerun for each of the 64 different combinations of ROG and NO_x emission control described above (see Figure 3 of ref *23*). These percentage reductions are applied "across-the-board" to all anthropogenic sources in the entire modeling region. The relative spatial distribution of anthropogenic emissions is not changed. The results of this simulation are summarized in Figures 2 and 3.

In Figure 2, the contours represent the number of days per year with ozone concentrations that equal or exceed the level of the 1- or 8-h U.S. Federal O_3 standards (0.125 and 0.085 ppm cutoffs, respectively, using the U.S. EPA's rounding conventions) at widely varying levels of anthropogenic ROG and NO_x control. The form of the 8-h standard involves the 3-year average of the annual fourth-highest daily maximum. Thus, assuming that 1987 represents an average year, violation of the standard corresponds to 4 days with O_3 concentrations greater than or equal to 0.085 ppm. In Figure 3, the contours show how the second-highest 1-h average O_3 concentration [ppb] and the fourth-highest 8-h average O_3 concentration for the entire year respond to ROG and NO_x emissions reductions. The upper pair of Figures 2 and 3 (panels a and b) in each case is based on the O_3 concentration predictions that occur anywhere within the boundaries of the SoCAB (see Figure 1) while the lower pair of Figures 2 and 3 (panels c and d) is based only on those O_3 concentration values that are predicted at the exact locations of the 34 ozone monitors in the SoCAB in 1987 (see ref *10* for monitoring site locations). The upper right corner of each plot represents historical 1987 emissions. The number of days in violation of the 1-h O_3 standard in 1987 is predicted to be 181 as compared to 172 such days actually observed; 249 days are predicted to exceed the concentration level of the 8-h O_3 standard as compared to 216 days actually observed.

Compliance with the 1-h Federal standard for O_3 is not predicted to be attained at any combination of anthropogenic emissions reductions studied. However, much improvement can be made when it is compared to the Federal 1-h standard. Moving across the top of panels c and d in Figure 2, we see that 60% ROG control (i.e., 40% of baseline ROG emissions remaining) with no further NO_x control relative to base case conditions would produce between 40 and 60 days per year with predicted 1-h average O_3 at monitoring sites greater than or equal to 0.125 ppm accompanied by about 120 days per year with predicted 8-h average O_3 at monitoring sites greater than or equal to 0.085 ppm. At 85% ROG control with no further NO_x control, fewer than 20 days per year would be predicted to equal or exceed 0.125 ppm O_3 over a 1-h averaging time at SoCAB monitoring sites, but the number of days per year with predicted 8-h average O_3 at or above 0.085 ppm never falls below 60 days per year anywhere within the range of the emissions control combinations studied. At

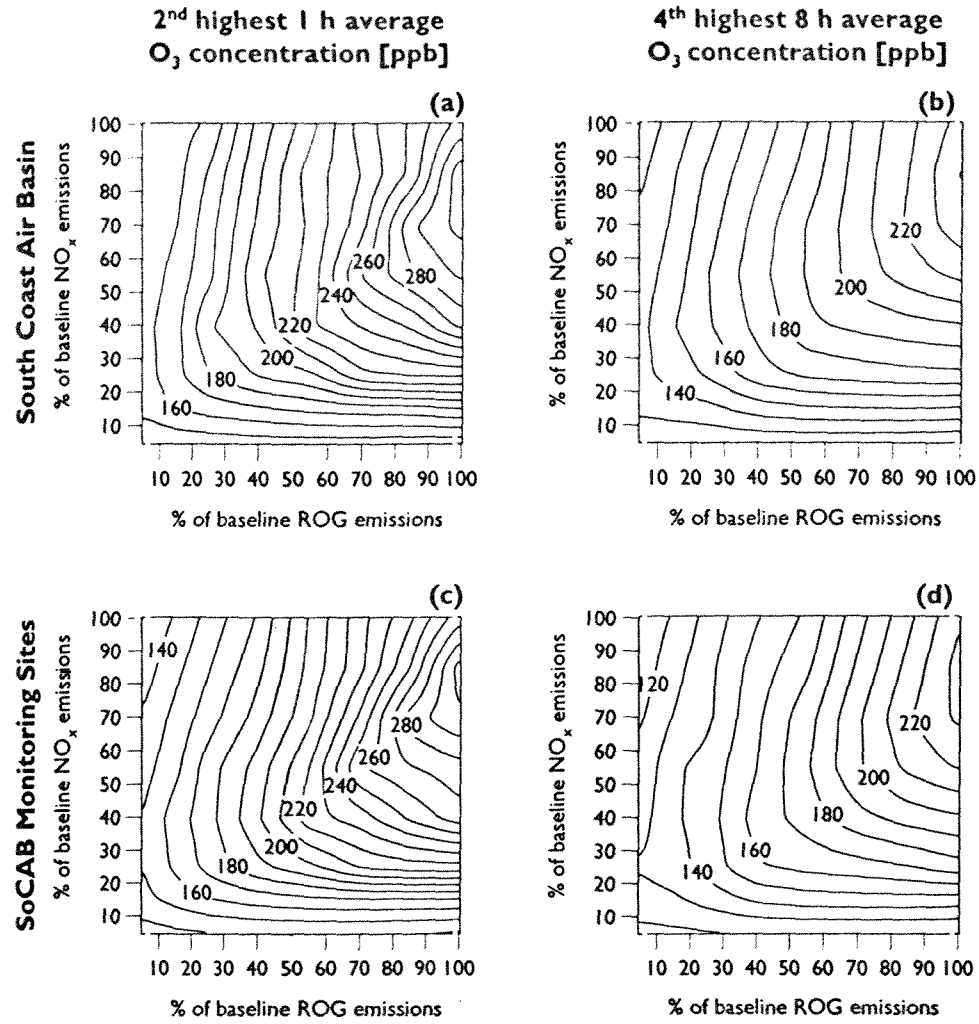

FIGURE 3. Isopleth diagrams showing the response of the predicted second-highest 1-h (panels a and c) and fourth-highest 8-h (panels b and d) average ozone concentrations for 1987 to alternative levels of basin-wide anthropogenic ROG and NO_x emissions in southern California remaining after control. Panels a and b are based on concentrations anywhere within the SoCAB; panels c and d are based on values predicted at SoCAB air monitoring sites.

least some NO_x control is needed to abate the high particulate nitrate concentrations in the southern California atmosphere (*24, 25*). As NO_x emissions are reduced at high levels of ROG control, the ozone control problem becomes more difficult. For example, 85% control of anthropogenic ROG and 45% NO_x control are predicted to reduce the frequency of violation of the Federal 1-h average O_3 standard at SoCAB air monitoring sites to 20 days per year under 1987 meteorological conditions. In contrast, 103 days per year are predicted to equal or exceed the level of the new Federal 8-h ozone standard at SoCAB monitoring sites at that same at high level of ROG and NO_x control. Even at very high levels of control, enough anthropogenic emissions remain to increase the 8-h average O_3 concentration within the SoCAB to 0.085 ppm or greater on many days per year when combined with biogenic emissions and the pollutant inflows present in the background air entering the air basin from upwind.

Another key result of this analysis is that the days that are most difficult to control are not necessarily the days that produced the highest peak ozone concentrations under 1987 historical conditions. At high levels of both ROG and NO_x control, the hardest days to bring below the O_3 standard often are those with the highest ROG emissions, which vary according to daily temperature conditions as discussed earlier. Higher than average temperatures over the western part of the SoCAB (e.g., Los Angeles and Orange Counties) that increase ROG emissions in the area with the highest density of emissions sources (i.e., highest traffic density) may be more important than peak temperatures inland. While the highest observed 1-h average ozone concentrations occurred in September and the highest 8-h average ozone concentrations occurred in July 1987, the hardest episodes to bring below the air quality standards through emissions control occurred over the October 2–6 period and possibly also over the April 20–23 period. All 20 days predicted to still exceed the 1-h O_3 standard at 85% RHC and 45% NO_x control were correctly predicted to exceed that standard during the base case historical year 1987 simulation. For the 103 days predicted to exceed the level of the 8-h O_3 standard at 85% RHC and 45% NO_x control, 80 days were correctly predicted to exceed the level of the standard during the base case historical year 1987 simulation.

Figure 4 shows the predicted shift in the frequency of occurrence of peak 8-h average O_3 concentrations as emission controls are applied. The 8-h average O_3 concentrations experienced on the highest concentration days of the year fall much more rapidly than the mid to low values due in part to the presence of ozone at a concentration of approximately 40 ppb in the background air advected into the region. These results are consistent with the analysis of measured ozone trends conducted by Lefohn et al. (*26*) that show that sites with the highest daily maximum 8-h average concentrations will respond to emission controls with a faster decrease in 8-h average ozone concentrations than is the

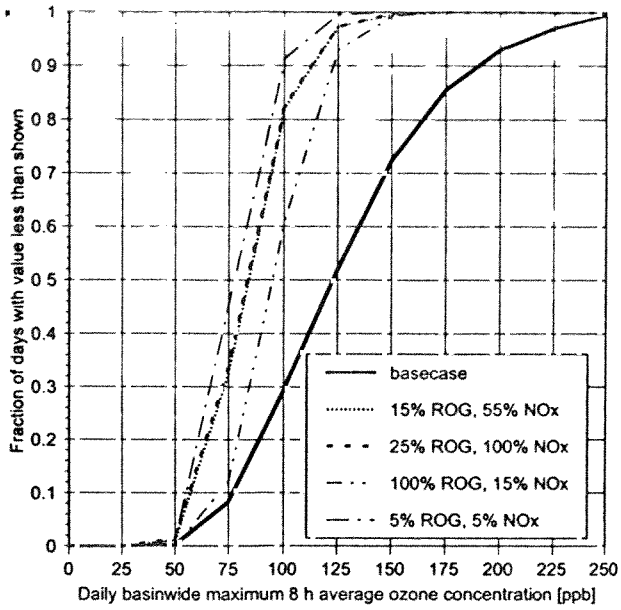

FIGURE 4. Cumulative frequency of occurrence of the daily regionwide maximum 8-h average ozone concentrations predicted anywhere within California's South Coast Air Basin for various levels of anthropogenic emissions remaining after control.

case at sites where the highest 8-h average concentration for the year is just above the 0.08 ppm level of the new standard.

Discussion

A method has been developed and demonstrated in which a photochemical airshed model driven by automated assimilation of routinely measured meteorological data can be used to predict how the frequency of occurrence of daily maximum 1- and 8-h average O_3 concentrations will shift in response to widely varying levels of emissions control. The simulations accurately predict the initial compliance status of the days that are predicted to remain above the air quality standards even with the most stringent emissions controls. Analysis showed that a program of stringent ROG and NO_x controls could reduce the frequency of occurrence of measured violations of the U.S. Federal 1-h average O_3 standard in southern California to perhaps 20–25 days per year. The same calculations showed that the level of the newly established 8-h average U.S. Federal O_3 standard will be exceeded more than 60 days per year, in some cases as high as 80–100 days per year, under 1987 meteorological conditions even at the most stringent levels of emission control considered here. This raises important public policy questions, as the U.S. Clean Air Act does not anticipate that it may be physically impossible for an airshed to attain compliance with the standards that have been set based on public health considerations.

It was found that the days with the highest measured O_3 concentrations in the Los Angeles area are not necessarily the days that are most difficult to bring below the national ambient air quality standards for O_3. This argues further that the process leading to the selection of an appropriate level of emission control for an air basin needs to consider conditions observed on more than a few historical days with the highest measured concentrations. Methods developed in the present study provide a way to examine and display results for essentially all days of the year when testing emission control strategy performance.

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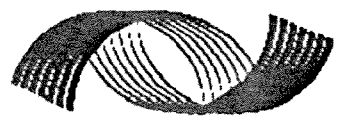

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Is EPA's Ozone Standard Feasible?

Randall Lutter

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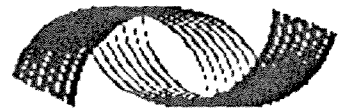
Randall Lutter is a resident scholar at the American Enterprise Institute and a fellow with the AEI-Brookings Joint Center for Regulatory Studies. He thanks Beth Mader for her valuable assistance and Rick Belzer, Bob Hahn and Sarah Reber for their helpful comments. His e-mail is rlutter@aei.org. The views expressed in this paper reflect those of the author and do not necessarily reflect those of the institutions with which he is affiliated.



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In response to growing concerns about understanding the impact of regulation on consumers, business, and government, the American Enterprise Institute and the Brookings Institution have established the AEI-Brookings Joint Center for Regulatory Studies. The primary purpose of the center is to hold lawmakers and regulators more accountable by providing thoughtful, objective analysis of existing regulatory programs and new regulatory proposals. The Joint Center builds on AEI's and Brookings's impressive body of work over the past three decades that has evaluated the economic impact of regulation and offered constructive suggestions for implementing reforms to enhance productivity and consumer welfare. The views in Joint Center publications are those of the authors and do not necessarily reflect the views of the staff, council of academic advisers, or fellows.

Executive Summary

The Environmental Protection Agency's estimate of the cost of meeting the new health-based ozone standard is likely to underestimate substantially the actual cost. EPA's cost estimates unrealistically assume that pollution control costs are capped at $10,000 per ton. Yet the required emission reductions in some cities exceed total motor vehicle emissions.

By dropping EPA's assumption that control costs are constant, I show that meeting the standard in 2010 would cost nearly $5 trillion in one city, and $70 billion in seven other cities. These cost estimates exceed EPA's estimates of $10 billion per year by orders of magnitude. I also find that the incremental costs of control are likely to far exceed any estimates of incremental benefits.

The high cost of meeting the ozone standard strongly suggests that it is likely to be infeasible in several cities. To avoid having EPA set such infeasible standards, Congress should amend the Clean Air Act to require the agency to balance the benefits and costs of regulation.

Is EPA's Ozone Standard Feasible?

Randall Lutter

Environmentalists often contend that statutes should allow or require regulatory agencies to issue rules to protect the environment without regard to the cost of such protection.[1] They have argued this point so successfully that regulatory agencies and courts have interpreted statutory provisions that are silent on the role of cost to prohibit consideration of cost in regulatory decisions. These statutes include the act establishing Superfund,[2] the Clean Water Act, and the Resource Conservation and Recovery Act. Perhaps the most notable such statute is the Clean Air Act, which directs the Environmental Protection Agency (EPA) to set air quality standards to "protect public health" with an "adequate margin of safety".[3] EPA and the courts have interpreted this language to prevent *any* consideration of cost.[4]

The neglect of cost in regulatory decision-making has given some key policy debates an Alice-in-Wonderland quality. In the case of the 1997 air quality standards, EPA Administrator Carol Browner rejected any consideration of cost until their implementation,[5] although there appears to be little flexibility at implementation because the Clean Air Act establishes specific deadlines by which compliance is mandatory.[6] This rejection was particularly important for the ozone standard, which EPA estimated had annual costs billions of dollars greater than likely benefits.[7] The statutory prohibition on considering costs thus provides a legal rationale for a policy position that otherwise would be bizarre.

There is widespread misunderstanding of the cost of EPA's 1997 air quality standards despite the attention to the standards.[8] EPA's estimate of $48 billion per year is

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[1] See, for example, Browner (1997), "Costs of meeting the standards and related factors have never been considered in setting the national ambient air quality standards themselves...I continue to believe that this is entirely appropriate."

[2] The Comprehensive Environmental Response, Compensation and Liability Act of 1980.

[3] See Clean Air Act, Public Law 101-549,1970; 42 United States Code 7409.

[4] See EPA (1997a) for a discussion of earlier court cases. See also *American Trucking v. EPA,* 175 F.3d 1027 (DC Cir 1999). See Lutter and DeMuth (1999) for a discussion of *American Trucking.*

[5] See Browner (1997).

[6] See, however, Melnick (1990).

[7] See EPA (1997b).

[8] See, for example, Wald (1999), Sunstein (1999), and Bentley and Haffner (1998). In May 1999, the U.S. Court of Appeals for the District of Columbia Circuit blocked the standards based in part on a finding that

based on an assumption that emissions reductions amounting to 80 percent of the total cost can be achieved at a constant rate of $10,000 per ton.[9] But this assumption lacks any empirical basis. More importantly, it violates the principle of diminishing returns. Therefore EPA's cost estimates are likely to be too low by an amount that Shogren dubbed the "lost triangle".[10]

Misunderstanding of the cost of the standards results from both the neglect of cost in regulatory decision-making and EPA's campaign to persuade the public that the rules are reasonable. Since the standards are health-based, cost estimates are irrelevant during judicial review, and actual and potential litigants have not examined them. Public commenters did not examine EPA's cost estimates because EPA published estimates of the cost of attaining the standards only *after* the deadline for public comments had passed.[11] Finally, analysts and advocates alike have seen little need to reassess the estimates because EPA's estimates already imply that the standards are the most costly regulatory initiative of the decade.

Such misunderstanding should be expected; independent analysts are often skeptical of agency estimates of the costs and benefits of their regulations.[12] Agency estimates are generally not subject to scientific peer-review or to judicial review. No government body independent of the executive branch reviews agency estimates of regulatory costs and benefits.[13] Moreover, agencies' prospective cost estimates rarely coincide with retrospective estimates of the effects of regulatory actions.[14]

Yet the reliability of agency estimates of regulatory costs (and benefits) is important, because reliable estimates are necessary to satisfy the public's right to know the expected effects of regulatory actions. Agency estimates of regulatory costs and benefits are typically the only official government estimates and thus are the basis of

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they represented an unconstitutional delegation of power. In October it rejected a request for an *en banc* hearing. EPA Administrator Carol Browner has said that she will seek an appeal and the Supreme Court may hear the case. See Lutter and DeMuth (1999).

[9] See EPA (1997b, ES-12 and ES-13). These and other values are in 1990 dollars. Values expressed in 1998 dollars would be about 24 percent higher.

[10] See Shogren (1998). Of course, EPA disagrees. It writes "the $10,000 cost estimate for these reductions is intended to provide ample margin to account for unknown factors associated with future projects, and *may tend to overestimate* the final costs of attainment" (emphasis added), EPA (1997b, p. ES-9).

[11] See EPA (1996a) and (1997b).

[12] See Lave (1996).

[13] See Lutter (1999).

[14] See Harrington, Morgenstern, and Nelson (1999), and Lutter (1999).

recent efforts to satisfy the public's right to know. The Unfunded Mandates Act of 1996 requires regulatory agencies to estimate and report the costs of their regulatory decisions. In addition, Congress has directed the Office of Management and Budget to report on the costs and benefits of federal regulations, and its reports use agency estimates.[15] If unreliable cost estimates misinform the public about the merit of regulatory programs, these reform efforts will be ineffective.[16]

In this paper I reassess the expected costs of EPA's 1997 ozone standard by relaxing EPA's assumption that sufficient emissions reductions will be available at a cost of $10,000 per ton and reinterpreting the cost curves implicit in EPA's analysis.[17] I find that attainment of the standard is infeasible in one city, and that costs in other cities in 2010 are about seven times greater than EPA's national estimates, even after allowing for technological progress. I also show that the cost of meeting the standard is not likely to fall over time—increasing levels of economic activity will dominate cost declines driven by technological progress. Thus, in cities where attainment is feasible but expensive, it may not be sustainable. Finally, I assess a set of emission control measures that is broader than those considered by EPA and includes taxes and fees on motor vehicle use and traffic congestion. I show that such implementation strategies would lower the cost of meeting the ozone standard, but that the cost would still exceed EPA's estimates.

The next section of this paper presents an analysis of EPA's data. This analysis includes illustrations of cost curves for Los Angeles, cost estimates for eight metropolitan areas, and estimates of the rate of change of cost over time. The subsequent section discusses market-based implementation measures. The final section explores broader policy implications.

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[15] See Office of Management and Budget (1998). See also Hahn (1999) and Hopkins (1991).

[16] See Urdan (1999), for a discussion of recent legislative efforts to assign to the General Accounting Office responsibility for conducting benefit cost analysis of federal regulations. See also Lutter (1999) for a discussion of the merit of such ideas.

[17] I go beyond Shogren (1998) who did not provide an estimate of the size of the lost triangle, but limit my analysis to ozone. For particulate matter, an assessment of the importance of EPA's assumption that necessary emissions reductions would cost no more than $10,000 / ton is complicated because EPA's analytic approach is different than for ozone and involves other assumptions as well.

Analysis

The single biggest difficulty in estimating cost is the need to extrapolate beyond the range of available data–a problem identical to one encountered in assessing the risk from environmental hazards. In assessing environmental risks, such as those from ozone and particulate matter, toxicologists and epidemiologists typically estimate an association between the risk of disease or injury and exposure to a hazard, *at some level of exposure.* They then extrapolate this association to estimate risk at much lower doses or levels of exposure. Of course, the extrapolation makes such estimates controversial.[18] In estimating the cost of meeting the ozone standard, I relate estimates of cost and emissions reductions for engineering-based control measures identified by EPA. These measures can achieve only a fraction of the emissions reductions needed to attain the standard in cities with serious air quality problems. Estimates of the cost of the standard must therefore involve extrapolations of cost curves well beyond the range of available data. Thus estimates of the cost of the ozone standard are subject to the same concerns about the validity of extrapolation as apply to low-dose risk assessments.[19]

In estimating the cost of meeting EPA's air quality standard, I take for granted all aspects of EPA's analysis with two exceptions. First, I relax its assumption that the cost of reducing emissions is capped at $10,000 per ton. Second, I allow explicitly for technological change and address uncertainties in future technological change by allowing for different rates of decline in marginal abatement cost.

EPA's estimates may be too low because they exclude indirect costs—a deficiency not addressed by this analysis. Indirect costs occur because efforts to remedy environmental problems can exacerbate distortions caused by pre-existing taxes. Economic research by Goulder and others indicates that indirect costs can be a large percentage of direct costs and may exceed them.[20] If estimates of indirect costs were added to the direct costs, the estimated total cost of meeting EPA's standard would be significantly higher than the estimates presented here.

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[18] See, for example, Ames and Gold (1996) and Hendee (1996).

[19] Of course there are other sources of uncertainty. The baseline from which emissions are reduced is uncertain because of uncertainties in future levels of economic activity and in the effectiveness of pending regulations to limit emissions. The emissions corresponding to attainment of the air quality standards are uncertain because the relation between emissions and air quality is relatively poorly understood.

[20] See Goulder, Parry, and Burtraw (1997), and Goulder and Williams (1999).

An Illustration: Los Angeles

I illustrate the analysis using EPA data from Los Angeles, Riverside and Orange counties, the area with perhaps the most severe ozone pollution in the country. Ozone is the product of chemical reactions involving sunlight and two sets of "precursors", volatile organic compounds (VOCs) and nitrogen oxides (NO_x). Since there are different control technologies for each precursor, I present cost information for each separately.[21] (See figures 1 and 2.)

EPA's data include cost and emissions reductions for a set of engineering measures to reduce emissions.[22] If these measures are ranked by their annual cost per ton of emissions reductions, they resemble a cost curve. In figure 1, I present EPA's data on the cost of controlling VOCs and a marginal cost curve fitted to these points. The baseline for the emissions reductions in figure 1 is a scenario for 2010 in which economic growth lifts emissions beyond current levels, but more stringent control measures limit emissions. Emissions reductions are measured in tons per day during the ozone season.

I estimate the marginal cost curve by assuming that the relationship between (the log of) marginal cost and (the log of) emissions reductions is quadratic. The fixed effect regression underlying this curve, presented in table 1, allows the slopes and the intercepts of the marginal cost functions to vary across the different metropolitan areas. As shown in table 1, almost all the coefficients in the cost functions are highly statistically significant and have the expected sign.[23]

The total cost of meeting the new standard is the area under the marginal cost curve between two levels of emissions. The first level corresponds to compliance with the old 1 hour standard that was issued in 1979, and the second level reflects attainment of the 8 hour standard issued in 1997. It is not clear from EPA's analysis that these levels of

[21] One control measure, a transportation control measure listed as "highway vehicles, gasoline", reduces 1.1 tons per day of VOCs and 2 tons per day of NO_x at a cost of 6.8 million dollars per year. As there is no simple way to incorporate measures with joint products into this analysis, I delete this measure from the scatterplots when estimating the cost functions, but calculate the emission reductions needed to meet the standard after netting out the emissions reductions achieved by this control measure.

[22] The data are in an Excel file Case1i.xls available in the EPA docket.

[23] The data used in the regressions are for a subset of the cities analyzed by EPA. The subset includes all cities for which the cost of meeting the standard is likely to be high. In particular, I select cities if the necessary emissions reductions beyond the identified measures are at least 100 tons per day for VOCs and NO_x combined and the reductions for each pollutant are greater than zero.

emissions correspond to the least-cost way of meeting the standard.[24] In particular, reducing NO_x emissions a little bit more and VOCs emissions a bit less may reduce the total cost of meeting the air quality standard. In the absence of information about such tradeoffs, I use EPA's estimates of the necessary emissions reductions. A recent scientific study indicates that this assumption is too optimistic; EPA's estimates of emissions reductions necessary to meet the standard are in fact too low.[25]

Rather than using a similar approach, EPA instead estimated cost by assuming that emissions reductions are available at $10,000 per annual ton per year, or about $4 million per ozone season daily ton. Figure 1 presents a graphical depiction of EPA's cost estimates as a rectangle.[26] The total cost of reducing VOC emissions to meet the 8 hour ozone standard in Los Angeles in 2010, from a baseline of attainment of the 1 hour standard, is about $340 million per year, according to EPA.

For NO_x, two estimates of the marginal cost of reducing emissions appear in figure 2. Curve a is the marginal cost derived from a total cost regression that is cubic in the log of emissions reductions.[27] Curve b, derived as for VOCs, is from the regression that appears in table 1. Interestingly, curve b appears not to fit the most expensive data points. Curve a, which appears to fit the Los Angeles data better than cost curve b, implies cost in 2010 of about $0.1 septillion for Los Angeles. Similar curves, however, do not fit data for other cities as well as the marginal cost functions that I choose to emphasize. EPA estimated the cost of reducing NO_x in Los Angeles to be only $580 million per year, an amount that is much less than the area under either cost curve a or b.

The cost curves presented in figures 1 and 2 do not take into account the technological change expected to occur before 2010—something that is obviously

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[24] See EPA (1997b).

[25] See Winner and Cass (2000).

[26] Since EPA did not estimate the costs of attaining the standards for each metropolitan area, I derive EPA's cost estimates from the reductions in tons of emissions per day during the ozone season necessary to attain the 8 hour standard, adjusted to annual tons based on relationships between the daily emissions reductions and the annual emissions reductions of the control measures EPA identified, and EPA's assumed cost of $10,000 per annual ton reduced. Excel File Case1i.xls, provided by EPA, indicates that the ozone season days per year implicit in the emissions estimates is 410 for NO_x and 329 for VOCs.

[27] The regression equation for NO_x emissions controls for Los Angeles is

$$total\ \cos t = \underset{(16.2)}{311} + \underset{(11.7)}{230}\,er - \underset{(2.82)}{54.6}\,er^2 + \underset{(0.225)}{4.36}\,er^3$$

where er denotes emissions reductions, both total cost and emissions reductions are transformed into natural logarithms, and the standard errors appear in parentheses.

difficult to forecast. Technological change among the identified technologies may be thought of as similar to the technological progress in manufactured goods. Research and development, learning-by-doing and human capital improvements will lower the cost of implementing these identified technologies, but the magnitude of such cost declines is hard to anticipate. Estimating the cost of implementing *unidentified* emissions reductions is even harder. When will research regarding these new technologies be completed? When will they be developed, marketed and adopted by industry, or mandated by regulators? What will be their cost and effectiveness when first adopted? There are no easy answers to these questions.

For simplicity, I account here for future technological change by estimating the rate of decline in the cost of emission controls based on the set of new technologies listed by EPA in its regulatory analysis. The arithmetic mean of the average rates of annual cost decline observed among the technologies cited by EPA is 7.7 percent.[28] An annual rate of change of 7.7 percent implies that cost in 2010, thirteen years after EPA's analysis was completed, would be about 37 percent of its 1997 value. This estimate overstates likely technological progress, however, because it reflects cost declines only in successful new technologies. Some new technologies, such as nuclear power generation, are adopted but then turn out to be more costly than originally anticipated. To account for such failures among new technologies I use 5 percent as an average rate of cost decline. In this case cost would equal 52 percent of their original values by 2010. The cost of meeting EPA's ozone standard in Los Angeles in 2010, based on rates of decline in abatement cost of 7.7 percent and 5 percent is $8.1 billion and $11.5 billion respectively.

How great would technological progress need to be in order for EPA's estimates of cost to be correct? The cost of abating emissions would have to fall by 27 percent per year from 1997 to 2010 for EPA's cost estimates to be correct. This is an extraordinarily and implausibly high rate of technological progress.

Estimates for Other Cities

For other cities the cost estimates are about $4700 billion in 2010, assuming that technological progress between 1997 and 2010 will lower cost by approximately half (see

[28] This estimate reflects only retrospective estimates of cost declines.

table 2.) All but $71 billion of the annual cost occurs in Fresno, California, where EPA estimates that NO_x and VOCs must be cut by more than 60 percent from baseline levels. But even ignoring the cost in Fresno, the cost estimates for the seven other cities—after netting out improvements due to technological progress—are seven times greater than EPA's cost estimate. For the New York City nonattainment area, the total cost of attaining the 8 hour standard, after technological progress, is $2.9 billion per year. For Washington-Baltimore, however, the expected cost is $7.4 billion per year, and for the San Francisco area the cost is $24 billion per year.

The true range of uncertainty about these cost estimates is quite large. The estimates are fairly sensitive to alternative assumptions about the form of the relationship between cost and emissions reductions, because they necessarily involve large extrapolations beyond the range of available data. Regressions not reported here tend, however, to give similar qualitative conclusions: costs are astronomical in a couple cities and generally many times greater than EPA's estimates.[29]

The broad conclusion—that the standard in some cities is too expensive to be met—is not surprising given that emissions reductions needed to meet it are very large relative to those available using identified measures. Table 3 shows estimates of the necessary emissions reductions, beyond the reductions from controls identified by EPA, as a percent of baseline emissions. Six cities require very large emissions reductions beyond those achieved by the identified measures: the reductions amount to more than 40 percent of baseline emissions. In addition, all eight cities require emissions reductions at least four times greater than the reductions from measures identified by EPA. For two cities, necessary reductions are more than ten times greater than those identified by EPA.

[29] Total cost functions that are quadratic in (the log of) emissions reductions implied that annual costs for two cities exceeded $1 trillion per year, for another city the cost exceeded $350 billion. The total cost for the remaining five cities I assessed was $19 billion per year. These estimates assume no technical progress. The regressions, however, have more coefficients that are statistically insignificant than the marginal cost functions presented here.

Changes Over Time

EPA suggests that technological progress may lower compliance cost over time.[30] But economic growth increases the emissions reductions necessary to meet the standard. Identifying the net change in cost over time is thus an unresolved empirical question.

The rate of change of total cost, as shown in the appendix, is a weighted average of the rate of change of the cost of controlling VOCs and the rate of change of the cost of controlling NO_x, where the weights are the share of total cost associated with each pollutant. The rate of change of the total cost, TC, of controlling each pollutant, g_{TC}, can in turn be estimated as

$$(1) \qquad g_{TC} = -\rho + g_L \left(\frac{\varepsilon_R \, TC(R)}{(R\ TC)/L} - \frac{\varepsilon_B \, TC(B)}{(B\ TC)/L} \right)$$

where ρ is the annual rate of decline in cost as a result of technological progress, g_L is the growth rate of laissez-faire emissions, and ε_R and ε_B are the elasticities of total cost with respect to emissions reductions evaluated at emissions reductions sufficient to meet the new standard, R, and to meet the old standard, B, respectively. The variables TC(R) and TC(B) are the total costs of achieving emissions reductions R and B from a baseline of zero reductions, and TC is the total cost of meeting the standard from a baseline of attainment of the old standard.

What do we know about the values of the parameters that determine the rate of growth of total cost? As described above, a good estimate of the rate of decline in cost is 5 percent per year. The rate of growth in uncontrolled emissions, g_L, is hard to estimate empirically in a regulated world but it is related to the rate of growth in real economic activity. Over the period from 1980 to 1997, GDP has grown at about 2.7 percent annually.[31] About a third of emissions comes, however, from the use of motor vehicles, which grows at the rate of about 2 percent per year.[32] Thus a good estimate of g_L is a weighted average of motor vehicle and other emissions, where the weights reflect the shares of total emissions. This weighted average is about 2.5 percent.

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[30] See EPA (1997a).
[31] See Council of Economic Advisers (1999, Table B.2).
[32] See EPA (1997b).

The variables ε_B and ε_R can be derived from the cost curves presented in table 1.[33] For Los Angeles, the elasticities for NO_x are 4.4, while for VOCs, ε_B and ε_R are 3.3 and 3.2 respectively.

Using these estimates, and the values of B, R, and L implicit in figures 1 and 2, implies that the cost of controlling NOx (to the level of the new ozone standard) grows 15 percent per year, while the cost of meeting the VOC standard (to the level of the new ozone standard) grows at 3 percent per year.[34] The cost of meeting the ozone standard in Los Angeles therefore will grow at a weighted average of these two estimates, or 12 percent per year, after 2010.

The rate of technological progress will have to be 20 percent per year in order for the cost of meeting the standard in Los Angeles to fall after 2010, based on the preceding estimates and equation (1). Such technological progress appears extremely unlikely.

Rapid growth in the cost of meeting the ozone standard does not imply that the present value cost of attaining the standard in perpetuity is infinite. That would be the case if annual cost continued to rise by more than the discount rate. But rising control cost would eventually curtail economic growth.[35] Indeed, rising costs suggest that cities that are able to attain the standard in 2010 may later find that attainment is too costly to be feasible.

Improved Implementation Strategies

The preceding cost estimates, like EPA's own estimates, ignore control measures based on behavioral changes—such as gasoline taxes and carpool programs—which some analysts believe can substantially reduce the cost of meeting EPA's air quality standard.[36] In fact, while such strategies may be much more efficient than some of the control measures analyzed above, they do not alter the basic conclusions of this paper.

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[33] The use of this baseline, which may differ from the conceptually correct baseline, imparts no clear bias to the results. Equation (1) uses a laissez-faire baseline, while the cost curves presented in figures 1a and 1b use a baseline of full-compliance with technology based requirements of the Clean Air Act. The uncertainty associated with identifying laissez-faire emissions in a regulated world makes the conceptually correct baseline unworkable.

[34] For NO_x, B and L and R are 318, 1048 and 460 respectively. For VOCs, the values are 526, 1064, and 628 respectively.

[35] See Aghion and Howitt (1998).

[36] See EPA (1999).

A study sponsored by the state of California reports that emissions control measures based on relatively small behavioral changes can be a cost-effective means of reducing local air pollutants (see table 4). Congestion pricing, for example, lowers congestion and greenhouse gas emissions in addition to reducing local air pollution, without an obvious effect on automobile accidents.[37] Since traffic delays are very costly, a reduction in them would offer large benefits that would likely offset other costs of such policies.[38]

But the low-cost emissions reductions resulting from such innovative implementation strategies are small compared with the reductions needed to attain the ozone standard. The California report indicates that congestion fees above the levels in table 4 might not be economically justified.[39] A $0.50 per gallon gasoline tax would achieve only a 4 percent reduction in NO_x emissions from motor vehicles; but even with a tax of $2.00 per gallon the reduction in motor vehicle emissions is only 12 percent.[40]

In fact, the complete *elimination* of emissions from motor vehicles appears insufficient to attain the standard in 2010 in some places. In Los Angeles, for example, the necessary emissions cuts beyond the measures identified by EPA are 33 percent for VOCs, but only 20 percent of VOCs come from motor vehicles.[41] In San Francisco, the NO_x emissions deficits is 45 percent, but motor vehicles contribute only 43 percent of baseline NO_x emissions.[42]

Conclusion

EPA's estimate of the cost of its ozone standard is much too low. In one city the cost is more than a trillion dollars per year while in seven others the costs total $70 billion per year, or about seven times EPA's estimate. Attainment of the standard appears infeasible by 2010.

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[37] With less congestion average vehicle speeds rise, but it is unclear whether this would increase or decrease the social cost of vehicle accidents.
[38] See Edlin (1999) and Calfee and Winston (1998).
[39] See the California Air Resources Board (1996, Table 7.6).
[40] Ibid, Table 7.8.
[41] See California Air Resources Board (1999). Emissions projections for 2010 from CARB are not identical to EPA's 2010 emissions forecasts. Thus CARB's estimates of the percent of total emissions that come from motor vehicles may be different than EPA's estimates; however, EPA's estimates of emissions from different sources are not publicly available.
[42] Ibid.

Attainment costs are likely to rise in years beyond 2010. Costs will rise because progress in new control technologies will be outweighed by increases in baseline emissions resulting from economic growth. Thus attainment of the ozone standard may later become infeasible in cities where meeting it in 2010 is simply very expensive.

The basic conclusion that the standard is infeasible is insensitive to changes in analytic methods. Of course, cost estimates based on extrapolations far beyond the range of available data are very uncertain. But in two cities attainment of the standard will require emissions reductions more than ten times greater than can be achieved by the emission control measures identified by EPA. In addition, the complete elimination of motor vehicle emissions would not ensure attainment of the standard in some cities.

This conclusion should be distinguished from similar conclusions based on new scientific information.[43] This analysis indicates that EPA knew or should have known that the standard was infeasible at the time it was issued. While infeasible standards always pose problems of public policy, such problems are especially acute when agencies issue infeasible standards and hide the infeasibility from the public.

Realistically, costs will never reach the trillions or even hundreds of billions of dollars per year implied by this analysis. Instead the managers of EPA's clean air programs and representatives of States will find new flexibility to avoid attainment of the ozone standard. For example, under the Clean Air Act, the State of California has responsibility for developing State implementation plans that EPA approves and for enforcing emissions limits on polluters that contribute to violations of air quality standards. If the annual cost for a metropolitan area indeed reached the tens of billions or more, affected States would simply get extensions and waivers from EPA, Congress and the courts.

Nevertheless, efforts to attain the ozone standard can still lead to costs that are excessive relative to the health and environmental benefits. EPA gave an upper bound estimate of the national benefits of the ozone rule of $8.5 billion per year,[44] but estimates of benefits consistent with the health effects estimated in the risk assessment blessed by EPA's Clean Air Scientific Advisory Council are hundreds of millions of dollars at

[43] See, for example, Winner and Cass (2000).
[44] See EPA (1997b, ES-16).

best.[45] These benefits estimates imply that a very generous upper bound for the benefits of controlling ozone is $10,000 per ton, and a more plausible value is a very small fraction of this estimate.[46] The upward slope to the marginal cost curves in figures 1 and 2 indicates that the last increment of emissions controls to reduce ozone is likely to have costs hundreds or thousands of times greater than any estimate of projected benefits.

With respect to regulatory issues more broadly, the ability of EPA to present erroneous estimates of the cost of a rule as important as its ozone standard should shed light on the merit of some regulatory reform initiatives that are popular in Washington. Initiatives that seek to increase the importance of agencies' estimates of the cost of their own regulations, without first ensuring the reliability of such estimates, can contribute to the public's misperceptions.

Finally, this analysis indicates a new need for cost to be considered in setting air quality standards. After all, adopting standards more stringent than is feasible does not improve children's health or the environment. Such standards do not provide health benefits any greater than less stringent ones, and the existence of infeasible mandatory standards may contribute to distrust of governmental institutions. In addition, there is little sense in setting standards that would cost many times more than the value of the benefits. To ensure a frank discussion of the tradeoffs implicit in major policy decisions, Congress should amend the Clean Air Act to direct EPA to consider costs in setting air quality standards. In particular, it should direct Congress to balance costs and benefits in controlling air pollution.

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[45] See EPA (1996a).

[46] See Lutter and Wolz (1997) and Lutter and DeMuth (1999) for an argument that reductions in ozone increase human exposure to harmful ultraviolet radiation by so much that the expected health improvements may be nil.

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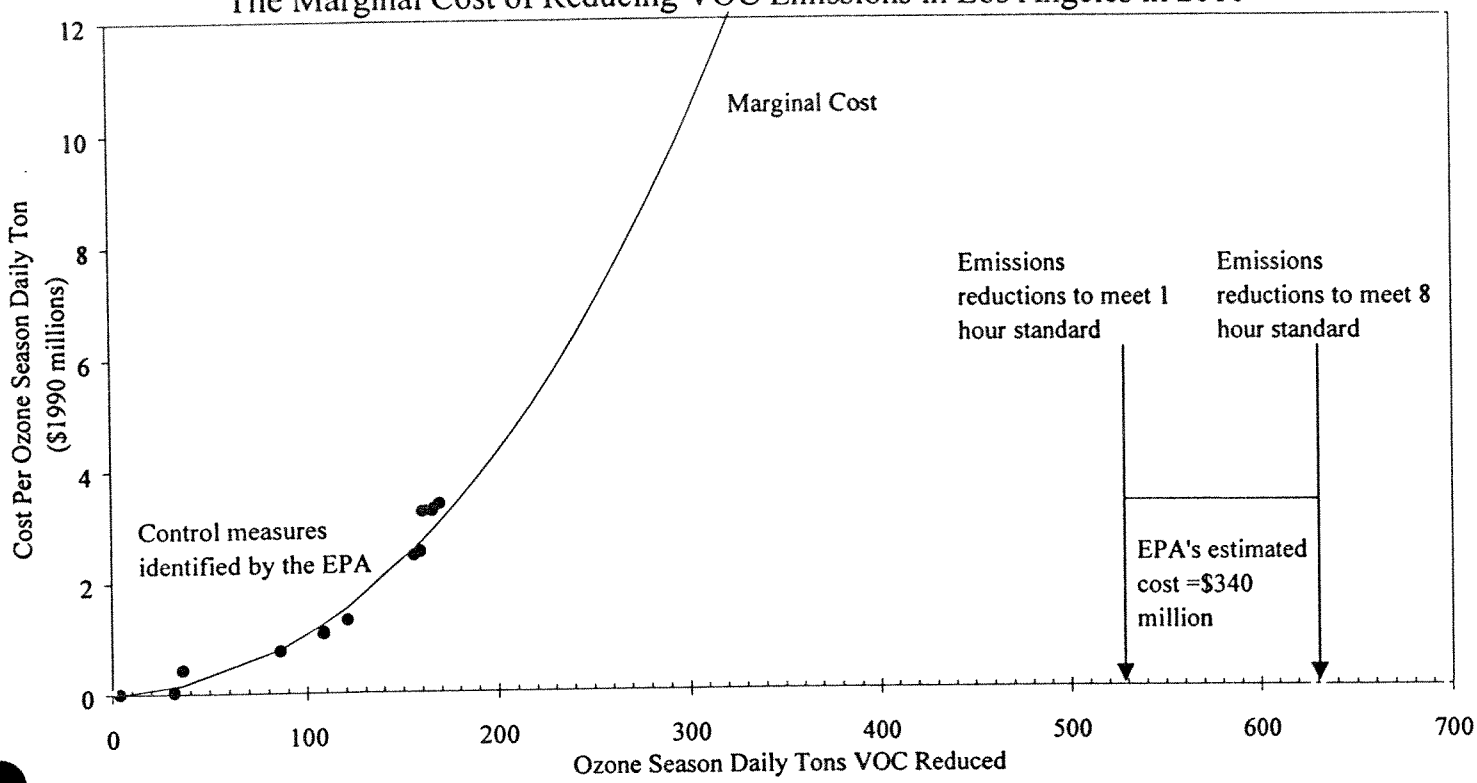
Figure 1

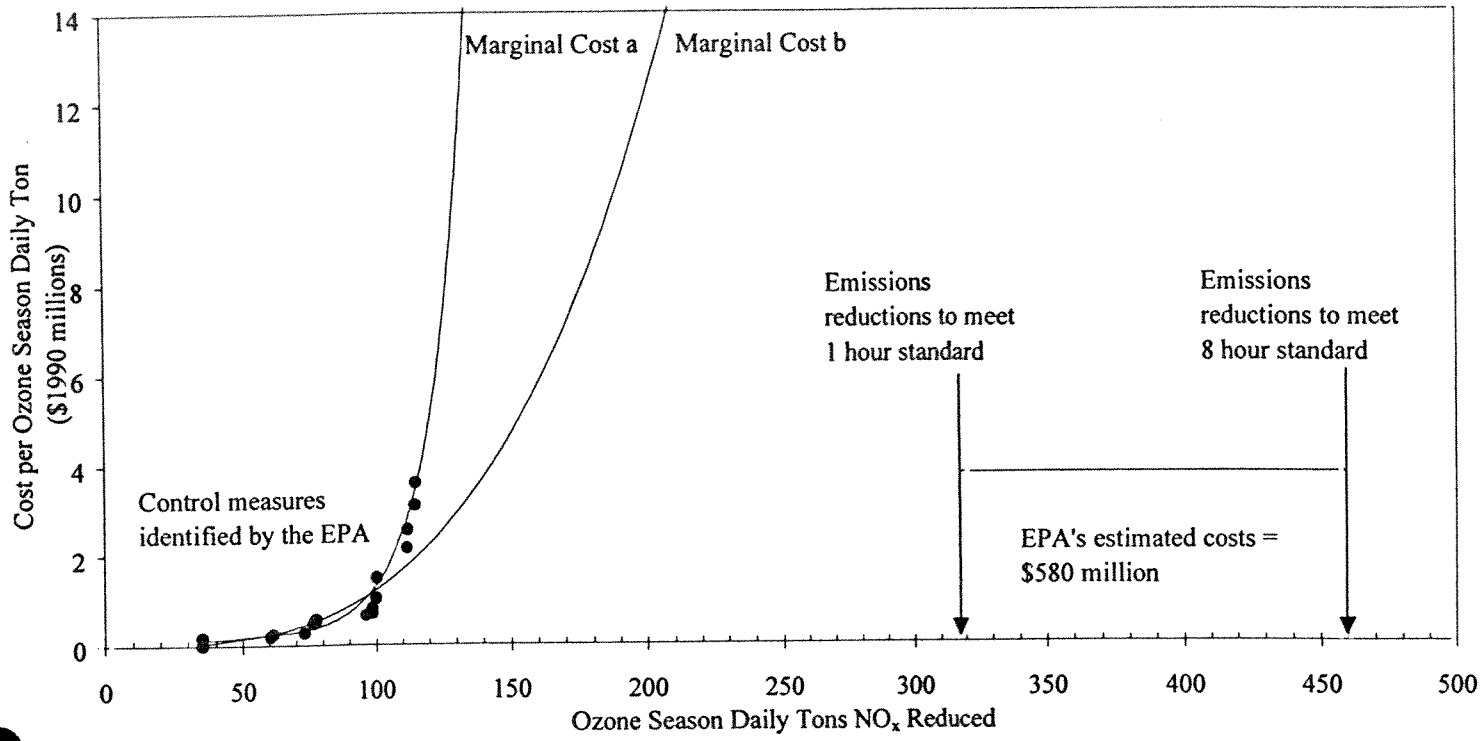
The Marginal Cost of Reducing VOC Emissions in Los Angeles in 2010



Note: The scatterplot excludes controls which reduce both NOx and VOCs and controls which have zero cost according to the EPA. The area described as Los Angeles includes Riverside, Orange and Los Angeles Counties. Emissions reductions are from EPA's baseline of 1054 tons per day. A cost per daily ton of $4 million is equivalent to a cost per annual ton of about $10,000. The benefits of controlling ozone are much less than $4 million per daily ton.

Figure 2

The Marginal Cost of Reducing NO_x Emissions in Los Angeles in 2010



Note: The scatterplot excludes controls which reduce both NO_x and VOCs and controls which have zero cost according to the EPA. The area described as Los Angeles includes Riverside, Orange and Los Angeles counties. Emissions reductions are from EPA's baseline of 1048 tons per day. A cost per daily ton of $4 million is equivalent to a cost per annual ton of about $10,000. The benefits of controlling ozone are much less than $4 million per daily ton.

Table 1
Marginal Cost Regressions for Eight Metropolitan Areas

Variables		Volatile Organic Compounds	Nitrogen Oxides
Square of Emissions Reductions		0.116 (.0432)	.0651 (.0257)
Bakersfield, CA	Intercept	10.6 (.338)	-15.0 (4.98)
	Emissions Reductions	1.35 (.169)	7.79 (1.41)
Fresno/Visalia-Tulare, CA	Intercept	10.6 (.336)	.716 (2.89)
	Emissions Reductions	1.01 (.175)	5.38 (1.28)
Los Angeles-Riverside-Orange	Intercept	7.43 (.829)	.362 (2.03)
	Emissions Reductions	.865 (.336)	2.67 (.498)
New York City-New Jersey-Long Island	Intercept	7.98 (.799)	12.5 (293)
	Emissions Reductions	.406 (.362)	.264 (.0734)
Philadelphia-Wilmington-Atlantic City	Intercept	8.39 (.503)	12.9 (.329)
	Emissions Reductions	.713 (.269)	.354 (.757)
Sacramento-Yolo, CA	Intercept	10.4 (.332)	14.0 (.364)
	Emissions Reductions	1.38 (.189)	1.06 (.208)
San Francisco/Modesto/ Stockton-Lodi	Intercept	9.13 (.486)	4.48 (1.52)
	Emissions Reductions	.827 (.265)	2.41 (.468)
Washington-Baltimore	Intercept	8.14 (.575)	12.7 (.279)
	Emissions Reductions	.855 (.291)	.697 (.153)
R^2		0.9984	0.9979
Number of Observations		121	101

Note: The table presents coefficients and standard errors in parentheses. Control measures that reduce both VOC and NO_x, and measures with zero cost are excluded from the regressions. All variables are in natural logarithms.

Table 2
Costs of Meeting the Ozone Standard in Selected Metropolitan Areas in 2010
Billions of 1990 Dollars

Annual Technological Progress	5 Percent			7.7 Percent
Area	Cost of Meeting VOC Target	Cost of Meeting NO_x Target	Total Cost of Meeting the 8 Hour Ozone Standard	Total Cost of Meeting the 8 Hour Ozone Standard
Bakersfield, CA	5.3	15	20	14
Fresno-Visalia-Tulare, CA	1.6	4700	4700	3300
Los Angeles-Riverside-Orange	2.4	9.1	12	8.1
New York-New Jersey-Long Island	0.97	1.9	2.9	2.0
Philadelphia-Wilmington-Atlantic City	0.68	0.69	1.4	0.96
Sacramento-Yolo, CA	0.71	2.8	3.5	2.5
San Francisco-Modesto-Stockton-Lodi	1.3	23	24	17
Washington-Baltimore	2.2	5.2	7.4	52
Total Excluding Fresno	14	57	69	50
Total	15	4700	4700	3300

Note: These cities are all those in EPA's data set with combined VOC and NO_x deficits greater than 100 tons per day and non-zero deficits for each pollutant.

Table 3
Meeting EPA's Ozone Standard Requires Very Large Emissions Cuts

Selected Cities	Reductions in 2010 to Meet the 8 Hour Standard Beyond the Identified Measures / Baseline Emissions (percent)		Reductions in 2010 to Meet the 8 Hour Standard / Reductions from Identified Measures	
	VOCs	NO_x	VOCs	NO_x
Bakersfield, CA	59	32	13	2.3
Fresno-Visalia-Tulare, CA	61	62	6.6	8.8
Los Angeles-Riverside-Orange	43	33	3.7	3.9
New York-New Jersey-Long Island	32	39	2.6	9.4
Philadelphia-Wilmington-Atlantic City	26	31	2.6	4.8
Sacramento-Yolo, CA	37	51	4.3	31
San Francisco-Modesto-Stockton-Lodi	28	45	3.1	7.1
Washington-Baltimore	41	39	3.3	8.8

Table 4
Alternative Emissions Control Measures in 2010
May Achieve Limited Emissions Reductions Cost-Effectively

Type of control measure	Tax rate	Metropolis	Other Benefits (percent reductions)				Previously Implemented?	Annual Government Revenue (millions)
			Traffic Delays	Emissions				
				Carbon	VOC	NO_x		
Congestion pricing	19 ¢ per mile	LA area	32	9.6	8.1	3.6	Not in U.S.	$7300
	13 ¢ per mile	SF Bay area	27	8.3	6.9	3.2		$2300
Minimum single driver employee parking fees	$1.00 /day	LA area	2.7	1.0	.8	.7	Not in U.S.	$1400
		SF Bay area	2.9	1.1	1.0	.9		$500
Gasoline tax	$0.50 / gallon	LA area	9.5	9.3	4.1	3.8	Yes	$3700
		SF Bay area	8.5	8.8	3.5	3.3		$1300
Mileage fee	2 ¢/ mile	LA area	11	5.2	4.2	3.9	Not in U.S.	$3100
		SF Bay area	9.0	4.1	3.8	3.6		$1100

Source: California Environmental Protection Agency, Air Resources Board, 1996. Note SF Bay area refers to the San Francisco Bay area and LA area refers to the Los Angeles area.

Appendix

Total costs can be written as

(1) $$TC = \int_{B(t)}^{R(t)} e^{-\rho t} MC(r)\, dr$$

where B(t) is the baseline emissions reductions, that is those that would occur in the absence of the air quality standards; t is an index for time, R(t) represents the emissions reductions necessary to meet the standard; $e^{-\rho t}$ is a factor accounting for cost declines related to technological progress assumed to occur at annual rate ρ; MC is the marginal cost of emissions reductions; and r indexes the amount of reductions. Applying Leibnitz's rule implies

(2) $$\partial TC / \partial t = -\rho TC + e^{-\rho t} MC(R)R' - e^{-\rho t} MC(B)B'$$

Using g_x to denote the growth rate of a variable x, (2) implies

(3) $$g_{TC} \equiv \frac{\partial TC}{\partial t}\frac{1}{TC} = -\rho + e^{-\rho t}\, MC(R)R'/TC - e^{-\rho t}\, MC(B)B'/TC$$

Given that R and B rise at constant rates, (3) simplifies to

(4) $$g_{TC} = -\rho + \varepsilon_R g_R \frac{TC(R)}{TC} - \varepsilon_B g_B \frac{TC(B)}{TC}$$

where ε_R and ε_B are the elasticities of total cost with respect to emissions reductions evaluated at emissions reductions R and B respectively, and TC(R) and TC(B) are the total costs of achieving emissions reductions R and B from a baseline of zero reductions. Note that emissions reductions B and R both grow over time because of growth in emissions under a laissez-faire policy. In fact, if laissez-faire emissions are L(t), then B' = R' = L'.

Since $g_R(R/L)=g_B(B/L)=g_L$, it follows that

(5) $$g_{TC} = -\rho + g_L \left(\frac{\varepsilon_R\, L\, TC(R)}{R\, TC} - \frac{\varepsilon_B\, L\, TC(B)}{B\, TC}\right)$$

Equation (5) is applicable to a cost curve for a single pollutant. Since both NO_x and VOCs contribute to ozone, I develop here an analogous expression applicable when there are two cost curves. Note that the total cost of meeting the ozone standard can be expressed as the sum of the costs of reducing VOCs and NO_x :

(6) $$TC = C_{VOCs} + C_{NOx}$$

Differentiating gives

(7) $$\frac{\partial TC}{\partial t} = \frac{\partial C_{VOCs}}{\partial t} + \frac{\partial C_{NOx}}{\partial t}$$

or

(8) $$g_{TC} = g_{CVOC}\frac{C_{VOCs}}{TC} + g_{CNOx}\frac{C_{NOx}}{TC}.$$

Thus, the growth rate for total cost is a weighted average of the growth rates for the cost of attaining the VOC target and the cost of attaining the NO_x target, where the weights are the shares of total cost attributable to VOCs and NO_x respectively.